Transport of labile carbon in runoff as affected by land use and rainfall characteristics

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Abstract

The mobilization of organic carbon (C) by water erosion could impact the terrestrial C budget, but the magnitude and direction of that impact remain uncertain due to a lack of data regarding the fates and quality of eroded C. A study was conducted to monitor total organic C and mineralizable C (MinC) in eroded materials from watersheds under no till (NT), chisel till (CT), disk till low input (DT-LI), pasture and forest. The DT-LI treatment relies on manure application and legume cover crops to partly supply the N needed when corn is grown, and on cultivation to reduce the use of herbicides. Each watershed was instrumented with a flume and a Coshocton wheel sampler for runoff measurement. Carbon dioxide (CO_2) evolved during incubation (115 days) of runoff samples was fitted to a first-order decomposition model to derive MinC. Annual soil ($6.2\,Mg\,ha^{-1}$) and organic C ($113.8\,kg\,C\,ha^{-1}$) losses were twice as much in the DT-LI than in the other watersheds ($<2.7\,Mg\,soil\,ha^{-1}$, $<60\,kg\,C\,ha^{-1}$). More than management practices, rainfall class (based on intensity and energy) was a better controller of sediment C concentration and biodegradability. Sediment collected during the low-intensity (fall/winter) storms contained more organic C ($37\,g\,C\,kg^{-1}$) and MinC (30–40% of sediment C) than materials displaced during the high-intensity summer storms ($22.1\,g\,C\,kg^{-1}$ and 13%, respectively). These results suggest a more selective detachment and sorting of labile C fractions during low-intensity storms. However, despite the control of low-intensity storm on sediment C concentration and quality, increased soil loss with high-energy rainfall suggests that a few infrequent but high-energy storms could determine the overall impact of erosional events on terrestrial C cycling.

1. Introduction

Water erosion is a major contributor to carbon (C) redistribution over terrestrial landscapes and export into aquatic systems. Mobilization of terrestrial C during erosional events could have measurable impact on the global C cycle, but quantitative assessments of the magnitude and direction of that impact are lacking. While some (Van Noordwijk et al., 1997; Stallard, 1998; Smith et al., 2001; McCarty and Ritchie, 2002) contented that retention of eroded C in terrestrial deposits and in aquatic systems (e.g. lakes and reservoirs) could lead to C sequestration, data from others (Gregorich and Anderson, 1985; Anderson et al., 1986; Beyer et al., 1993) indicated that materials entrapped in terrestrial deposits contain labile C fractions which, under aerobic conditions, could undergo mineralization. Beyer et al. (1993) reported that up

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to 70% of SOC in eroded soil could be decomposed during transport and deposition. To reconcile these divergent findings, an evaluation of the quality of C transported in overland flows is proposed.

In soils, organic C contributes to the formation and stability of aggregates which in turn provide an environment for the physical protection of otherwise labile C fractions against microbial degradation. The stability of soil aggregates is an important soil property which is related to total soil C pool, water-extractable carbohydrate (Haynes and Swift, 1990) and soil mineralogy (Zech et al., 1997). Aggregate stability generally increases with a reduction in tillage intensity (Mahboubi et al., 1993). Beare et al. (1994) reported significantly greater amounts of aggregate-protected C in no-till than in plowed soils. Research has shown that the resistance of soil aggregates to external forces is inversely related to their size, and that the least-resistant macroaggregates generally contain a greater proportion of the labile C (Angers and Giroux, 1996; Jastrow et al., 1996) and recently deposited C (Puget et al., 1995). This spatial arrangement indicates that a major impact of water erosion will be the release of labile C associated with the larger soil aggregates.

A rainfall simulation study (Jacinthe et al., 2002) conducted to assess the pool of mineralizable C in runoff from no-till and plow-till microplots ($\sim$1 m^2) showed that 29–35% of the runoff C is potentially mineralizable, demonstrating the preferential removal of the labile C fractions from soils by water erosion. The results also showed that the pool of mineralizable C was significantly greater in runoff from no-till than from plow-till soils, suggesting that the quality of runoff C reflects that of soil C at the site from which runoff originates. It should be recognized however that mesocosm study results may not adequately describe a scale-dependent process such as soil erosion (Evans, 1995). Further, when using mesocosm studies, it is neither possible to replicate the hydrodynamic processes (soil detachment, deposition and resuspension) which characterize water erosion, nor capture the impact of landforms distribution, management practices and season on the biological quality of eroded C.

A watershed-scale investigation of the bioavailability of runoff C was conducted to validate the data obtained from the rainfall simulation study. The field study was designed to determine whether an association exists between runoff C quality and C quality of soils from where runoff originates. Specifically, the objectives of that investigation were to (i) assess the pool of total and mineralizable C in sediments from watersheds under different land use and management practices and (ii) determine the effect of management and season on the concentrations of labile C in runoff.

2. Materials and methods

2.1. Site description

The study was conducted from May 2001 to May 2002 at the North Appalachian Experimental Watersheds (NAEW) near Coshocton, OH (40°22′N, 81°48′W; elevation: 300–430 m) in the unglaciated section of the state of Ohio. Long-term (60 years) mean annual temperature is 10.4 °C and rainfall is 950 mm. September is the driest, and June and July are the wettest months of the year (Kelley et al., 1975).

Soils are well-drained silt-loam developed from shale and sandstone bedrocks (Kelley et al., 1975). Included in this study were (i) a no-till (WS 118), (ii) a chisel-till (WS 123), (iii) a disk-till (WS 127), a pasture (WS 129) and (iv) a forest (WS 172) watersheds. The no-till (NT) and chisel-till (CT) watersheds were in a 2-year corn (*Zea mays* L.)–soybean (*Glycine max* L.)/rye (*Secale cereale* L.) rotation (rye used as a winter cover crop following soybean) since 1984, and received fertilizer (225 kg N ha^{-1} per year during the corn year) and herbicides at recommended rates. The CT watershed was chiseled to a depth of 25 cm each spring. Both NT and CT watersheds were planted to corn during the study period. The disk-till watershed (WS 127) was in the soybean phase of a 3-year corn–soybean–wheat (*Triticum aestivum* L.)/clover (*Trifolium pratense* L.) rotation that began in 1990. In the spring of 2001, WS 127 was disked three to four times prior to planting soybean. As this watershed receives less agro-chemicals (fertilizer, herbicide) than the other watersheds, it is referred to as disk till low input (DT-LI) treatment. In this rotation, winter wheat was drilled into the soybean residue in the fall, and clover was broadcast seeded into the standing wheat and allowed to grow until the following spring. The clover residue along with cattle manure (5–9 Mg ha^{-1}, <70% water content) was incorporated into the soil to supply some of the nitrogen needed by the next

Table 1
Watershed characteristics and annual soil loss

	Watershed				
	118	123	127	129	172
Land use	Cropland	Cropland	Cropland	Pasture	Forest
Vegetation[a]	Corn	Corn	Soybean	Grass	Trees
Area (ha)	0.79	0.55	0.67	1.1	17.7
Slope range (%)	6–12	2–12	6–18	12–25	18–35
Dominant soil type[b]	Co, Cl	Ke, Re	Co	Be	Be, Co
Average runoff volume (mm)					
Class 1	1.8	5.8	4.7	0.14	3.2
Class 2	1.3	2.8	4.2	–	5.2
Class 3	13.6	17.9	21.1	5.5	13
Sediment concentration ($g\,l^{-1}$)					
Class 1	0.5	0.4	1	0.4	0.2
Class 2	1.3	0.5	0.9	–	0.2
Class 3	3.2	3.7	3.7	0.3	0.4
Erosion rate ($Mg\,ha^{-1}$ per year)	2.72	2.9	6.23	0.03	0.01
Carbon loss ($kg\,ha^{-1}$ per year)	59.9	41.3	113.8	3	0.5
DOC in runoff ($kg\,ha^{-1}$ per year)	12.5	11.6	19.6	2.1	0.3

[a] Vegetation during monitoring period.

[b] Co: Coshocton (fine-loamy, mixed, mesic Aquultic Hapludalfs); Cl: Clarksburg (fine-loamy, mixed, mesic Oxyaquic Fragiudalf); Ke: Keene (fine-silty, mixed, mesic Aquic Hapludalf); Re: Rayne (fine-loamy, mixed, mesic Typic Hapludult); Be: Berks (loamy-skeletal, mixed, mesic, Typic Dystrudept).

corn crop. The DT-LI watershed received half the recommended herbicide rate with most of the weed control achieved through mechanical cultivation two to three times during the growing season. All tillage and planting operations were performed along the contour. Vegetation at the pasture watershed (WS 129) consisted of orchard grass (*Dactylis glomerata*) and blue grass (*Proa pratensis* L.). The forest watershed (WS 172) was a mixture of oak (*Quercus* spp.) and pine (*Pinus* spp.). Additional information regarding land use history and soil properties are provided in Table 1 and elsewhere (Kelley et al., 1975; Shipitalo and Edwards, 1998; Owens et al., 2002).

2.2. Soil sampling

Soil samples were collected in May 2001 to assess physical and biochemical properties of soils in the different watersheds. Samples were taken at depths 0–10 and 10–20 cm at duplicate points in the summit, midslope and downslope locations in each watershed. Soil cores (5 cm diameter) were also taken near each sampling point to determine bulk density. Each composite soil sample thus collected was split into two portions.

A portion was air-dried and used for determination of soil pH, SOC (dry combustion) and aggregate stability (wet sieving; Yoder, 1936). The second portion that was kept field-moist (4 °C), sieved (<2 mm) and used in the assessment of soil respiration (monitoring CO_2 evolution during laboratory of incubation).

2.3. Runoff sampling

Each watershed is instrumented with recording rain gauge, runoff volume recorder and a sediment collection system consisting of a proportional runoff sampler (modified Coshocton wheels; Bonta, 2002) and a tank located in a refrigerated room near the watershed outlet. After a runoff-producing storm, the tank content was thoroughly mixed and a volume of approximately 1 l runoff sample was taken in acid-washed polypropylene bottles with care being taken to include both dissolved and suspended material in the sample. Once collected, runoff samples were shipped in ice to the laboratory in Columbus for processing.

Duplicate 300 ml runoff volume was transferred into beakers. Runoff was dried in a forced-air oven (105 °C; 2–3 days) and the mass of dissolved solids measured.

Sediment concentration was determined by filtration of runoff samples (Owens et al., 2002). Dried sediment was scrapped from beakers, ground, sieved (250 μm) and stored for C analysis. Aliquot (∼20 ml) of runoff sample was filtered through a Fisherbrand glass fiber filter (0.45 μm) and the filtrate used for dissolved organic C determination.

2.4. Assessment of mineralizable C in runoff

The procedure described in Jacinthe et al. (2002) was used in this assessment. Briefly, 40 ml runoff sample was transferred into duplicate 160 ml glass bottles that were subsequently stoppered, crimp-sealed and flushed with CO_2-free air. Bottles were placed in an incubator at 25 °C. Air samples were taken from the incubation bottle headspace at days 5, 12, 20, 40, 70 and 115 to determine CO_2 concentration. Incubation vessel was flushed with CO_2-free air after each sampling occasion following day 12. Total CO_2 production was computed as the sum of CO_2 in the gas and liquid phases (Jacinthe et al., 2002), and the data were fitted to a first-order kinetic model ($C_t = C_0(1-\exp(-kt))$), where C_t is the cumulative CO_2-C (g CO_2-C kg^{-1} sediment) evolved at time t (days), C_0 the potentially mineralizable C (g C kg^{-1} sediment) in runoff and k the rate constant (per day). The procedure NLIN (method: Marquardt) available in SAS was used.

2.5. Analytical procedures and computation

Analysis of runoff filtrate for DOC was performed in a C analyzer (Model 700; Oceanography International, College Station, TX) after treatment of the filtrate with H_3PO_4 to purge inorganic C. Finely ground (250 μm) soil and sediment samples were analyzed for total C by the dry combustion procedure using a C–N analyzer (Carlo-Erba NC 2100). Inorganic C concentration of sediment samples (0.5–1 g sediment) was determined through decomposition of carbonates with acid (2 ml of 4 M HCl; Bundy and Bremner, 1972) in a closed vessel and analysis of the CO_2 produced. Organic C was computed as the difference between total and inorganic C. The analysis of air samples for CO_2 was conducted using a gas chromatograph (Shimadzu GC-14A) equipped with a thermal conductivity detector (150 °C) and with helium (20 ml min^{-1}) as the carrier gas.

Sediment export was computed by multiplying sediment concentration (g sediment l^{-1}) and runoff volume (l). Carbon export during a rainfall event was computed as the product of soil loss (kg soil loss ha^{-1}) and sediment C concentration (g C kg^{-1} sediment). Annual export was computed by summing C export during all recorded storms.

Runoff was collected 31 times during the course of this study; however, runoff was generated only 4 and 11 times in the pasture and forest watersheds. Average and maximum rainfall intensities (mm h^{-1}), and rainfall energy, E (J m^{-2} mm^{-1} rainfall) were computed for each event leading to runoff collection (Table 2). Rainfall energy was computed using the model ($E = 11.9 + 8.78 \log I$, where I is the rainfall intensity in mm h^{-1}) proposed by Wischmeier and Smith (1958). Total rainfall energy (E_T) ranged between 22 and 2344 J m^{-2} with a geometric mean of approximately 200 J m^{-2} (Table 2). Thus, based on total energy, rainfall events were divided into (i) class 1, where $E_T < 200$ J m^{-2}, (ii) class 2, where $200 < E_T < 400$ J m^{-2} (i.e. $2 \times$ geometric mean), and (iii) class 3, where $E_T > 400$ J m^{-2}. Classes 1, 2 and 3 correspond, respectively, to (i) low amount and low intensity, (ii) moderate-to-high amount and moderate intensity, and (iii) high amount and high-intensity storms. The class 3 storms were mostly recorded during the summer months, whereas the class 1 events were more common during the late autumn and winter. For the cultivated watersheds, there were on average 4–11 measurements of the response variables (sediment C and biodegradability) within each rainfall class. Using this pseudo-replication approach, analysis of variance (ANOVA) was conducted using rainfall energy class and watershed as class variables and the general linear modeling (GLM) procedure available in SAS (SAS Institute, 1990). The non-cropped watersheds were excluded from this analysis as these watersheds produced runoff during a limited number of rainfall events. Statistical significance was determined at the 95% level.

3. Results

3.1. Watershed soil properties

No difference was detected between the NT (WS 118) and DT-LI (WS 127) watersheds with respect to

Table 2

Characteristics of the rainfall events during which runoff was collected and analyzed

Date	Rainfall (mm)	Mean intensity (mm h^{-1})	Peak intensity (mm h^{-1})	Rainfall energy[a] (J m^{-2})	Class[b]
21 May 2001	34.79	29.73	167.64	863	3
27 May 2001	14.99	9.67	106.68	307	2
2 June 2001	12.45	19.45	55.88	289	2
6 June 2001	8.13	5.2	27.43	147	1
15 June 2001	16.25	55.3	114.3	440	3
25 July 2001	90.17	41.26	114.3	2344	3
31 August 2001	24.13	10.74	59.06	504	3
19 November 2001	12.45	1.47	4.57	166	1
25 November 2001	16.51	20.67	187.96	386	2
27 November 2001	14.48	1.97	6.93	210	2
30 November 2001	12.19	2.78	14.07	193	1
8 December 2001	2.28	0.57	1.52	22	1
14 December 2001	6.85	0.84	2.54	77	1
24 January 2002	8.38	1.99	12.19	122	1
30 January 2002	10.92	0.68	2.29	114	1
1 February 2002	8.38	1.66	5.26	116	1
2 March 2002	13.21	1.58	6.77	180	1
26 March 2002	22.1	1.37	6.1	290	2
29 March 2002	5.33	2.08	10.16	78	1
1 April 2002	1.78	1.75	4.35	25	1
3 April 2002	8.13	1.8	7.62	115	1
13 April 2002	32.77	8.42	76.2	655	3
14 April 2002	39.37	10.72	76.2	823	3
19 April 2002	14.22	10.93	45.72	299	2
21 April 2002	6.35	0.98	3.05	75	1
27 April 2002	12.19	2.23	15.24	182	1
28 April 2002	15.24	14.74	45.72	337	2
8 May 2002	4.06	1.06	3.27	49	1
13 May 2002	18.03	1.29	5.08	233	2
14 May 2002	5.08	4.2	15.24	88	1
17 May 2002	14.98	1.14	2.81	186	1

[a] $E_T = (11.9 + 8.78 \log I)R$, where R (mm) and I (mm h^{-1}) are rainfall amount and intensity, respectively.

[b] Class 1: $E_T < 200$ J m^{-2}; class 2: $200 < E_T < 400$ J m^{-2}; class 3: $E_T > 400$ J m^{-2}.

SOC pools and depth distribution ($27.5_{(0-10\,cm)}$ and $15.9_{(10-20\,cm)}$ Mg C ha^{-1}; Table 3). In the CT (WS 123) watershed, SOC pool was slightly lower but less stratified ($22.9_{(0-10\,cm)}$ and $20.1_{(10-20\,cm)}$ Mg C ha^{-1}). Carbon stocks in the non-cropped watersheds were 1.5-fold higher (57–61 Mg C ha^{-1}) than in the cultivated watersheds. Mean-weighted diameter (MWD) of soil aggregates averaged 2.0 mm in the forest and pasture watersheds; in the cropped watersheds MWD was lower (range: 0.6–1.8 mm) and was in the order NT > CT = DT-LI. Land use and management practices had a net effect on soil C mineralization. During the 115 days incubation, C mineralization rates in the DT-LI samples were similar to rates (0.45–0.50 g C kg^{-1}) observed with the pasture watershed samples, and

were higher than those from the other cropped watersheds. Mineralized C during the incubation period represented 1–2% of the total SOC pool (Table 3).

3.2. Rainfall and sediment export

During the monitoring period (May 2001–May 2002), total rainfall amounted to 1037 mm. The relationship between concentrations (g l^{-1}) of sediment (y) and total dissolved solids (x) in runoff was $y = (0.78 \pm 0.03)x + 0.11$, $R^2 = 0.91$, $P < 0.001$. Significant effects of rainfall class (as defined by total rainfall energy) on runoff sediment concentration ($P < 0.02$) and sediment export rate ($P < 0.03$) were found.

Table 3
Selected properties of soils at the North Appalachian Experimental watersheds

WS	Tillage[a]	Rotation[b]	pH	Bulk density ($g\,cm^{-3}$)	MWD[c] (mm)	Organic C ($g\,C\,kg^{-1}$ soil)	MinC[d] ($g\,CO_2$-$C\,kg^{-1}$ soil)	Enrichment ratios[e]	
								SOC	MinC
Soil depth (0–10 cm)									
118	NT	CSR, 17 years	6.6	1.5 (0.1)	1.8	18.3 (0.1)	0.32 (0.00)	1.9	35.4
123	CT	CSR, 17 years	6.4	1.3 (0.1)	1.3	17.6 (0.0)	0.34 (0.01)	1.7	38.5
127	DT-LI	CSW/Cl, 11 years	6.9	1.5 (0.2)	1.6	18.4 (0.1)	0.45 (0.00)	1.6	17.9
129	Pasture	>55 years	6.1	1.5 (0.1)	1.8	28.6 (0.1)	0.50 (0.01)	2.9	41.6
172	Forest	>62 years	5.4	1.0 (0.2)	2.9	36.0 (0.7)	0.79 (0.02)	1.2	17.9
Soil depth (10–20 cm)									
118	NT	CSR, 17 years	6.5	1.5 (0.1)	1.2	12.0 (0.1)	0.11 (0.01)	2.9	102.9
123	CT-C	CSR, 17 years	6.6	1.5 (0.3)	1.1	13.4 (0.1)	0.19 (0.01)	2.2	109.1
127	DT-LI	CSW/Cl, 11 years	7	1.6 (0.1)	0.6	8.6 (0.1)	0.12 (0.01)	3.4	67.3
129	Pasture	>55 years	7.5	1.4 (0.2)	2	14.9 (0.1)	0.16 (0.01)	5.6	130
172	Forest	>62 years	5.2	1.3 (0.3)	1.1	15.8 (0.2)	0.13 (0.00)	2.8	108.8

[a] NT: no till; CT: chisel till; DT: disk till; LI: low chemical input.

[b] C: corn; S: soybean; R: rye; W: wheat; Cl: clover.

[c] MWD: mean weight diameter of soil aggregates.

[d] MinC: carbon mineralized after 115 days of incubation of soil (<2 mm soil aggregates).

[e] Eroded materials/soil aggregates.

The largest sediment-producing storms occurred on 21 May and 25 July 2001 and on 13–14 April 2002 (Table 2) with soil loss across cropped watersheds averaging 1.1, 2.3 and 0.5 Mg soil ha^{-1}, respectively, on these dates (Fig. 1). Total soil loss from the NT, CT and DT-LI watersheds during the study period amounted to 2.7, 2.9 and 6.2 Mg ha^{-1} per year, respectively (Table 1). Thus, the three most intense (out of a total of 31 events) rainstorms recorded during the study period accounted for 90–94% of the annual soil loss. During the 25 July storm, soil loss from the DT-LI watershed was five times higher (4.58 Mg ha^{-1}) compared to the NT (1.1 Mg ha^{-1}) watershed (Fig. 1). Sediment export from the non-cropped (pasture and forest) watersheds was <0.03 Mg ha^{-1} per year. A strong exponential relationship (Fig. 2a) was found between daily soil loss (y) and total energy of a storm (x).

3.3. Carbon export in runoff

Inorganic C concentration in runoff averaged 4.8 g C kg^{-1} sediment with no significant difference among the three cultivated watersheds (Fig. 1). A trend of diminishing inorganic C concentration with increasing rainfall energy was also noted (9.2, 3.9 and 0.5 g inorganic C kg^{-1} sediment for classes

1, 2 and 3 storms, respectively). Although sediment from the NT watershed contained on average more organic C (34.8 g C kg^{-1}) than sediment from the CT (30.1 g C kg^{-1}) and DT-LI watersheds (29.2 g C kg^{-1}), differences were not statistically significant (Fig. 1). The average C concentration (mean: 49.8; range: 14.0–100.7 g C kg^{-1}) of sediment from the non-cropped watersheds was higher than from the cultivated watersheds. Rainstorm type, however, had a significant effect ($P < 0.001$) on sediment C concentration which averaged 38.2, 35.1 and 22.1 g C kg^{-1} sediment for the classes 1, 2 and 3 rainstorms, respectively (Table 4). As the class 3 storms were more common during the summer, it follows that sediment C concentration (the pasture watershed, excepted) was lower during the summer (mean: 18 g C kg^{-1}) than in the other seasons. More C was exported from the DT-LI watershed (113.8 kg C ha^{-1} per year) than from any other watersheds (NT: 59.9; CT: 41.3; pasture: 3.0; forest: 0.5 kg C ha^{-1} per year) (Table 1). Annual C loss in runoff was strongly related (Fig. 2b) to sediment export.

Concentration of DOC in runoff samples ranged between 3.5 and 27.6 mg C l^{-1}. Although DOC appeared to vary seasonally (mean: 24.2 mg l^{-1} in the fall and 10.4 mg l^{-1} in the other seasons), no statistically
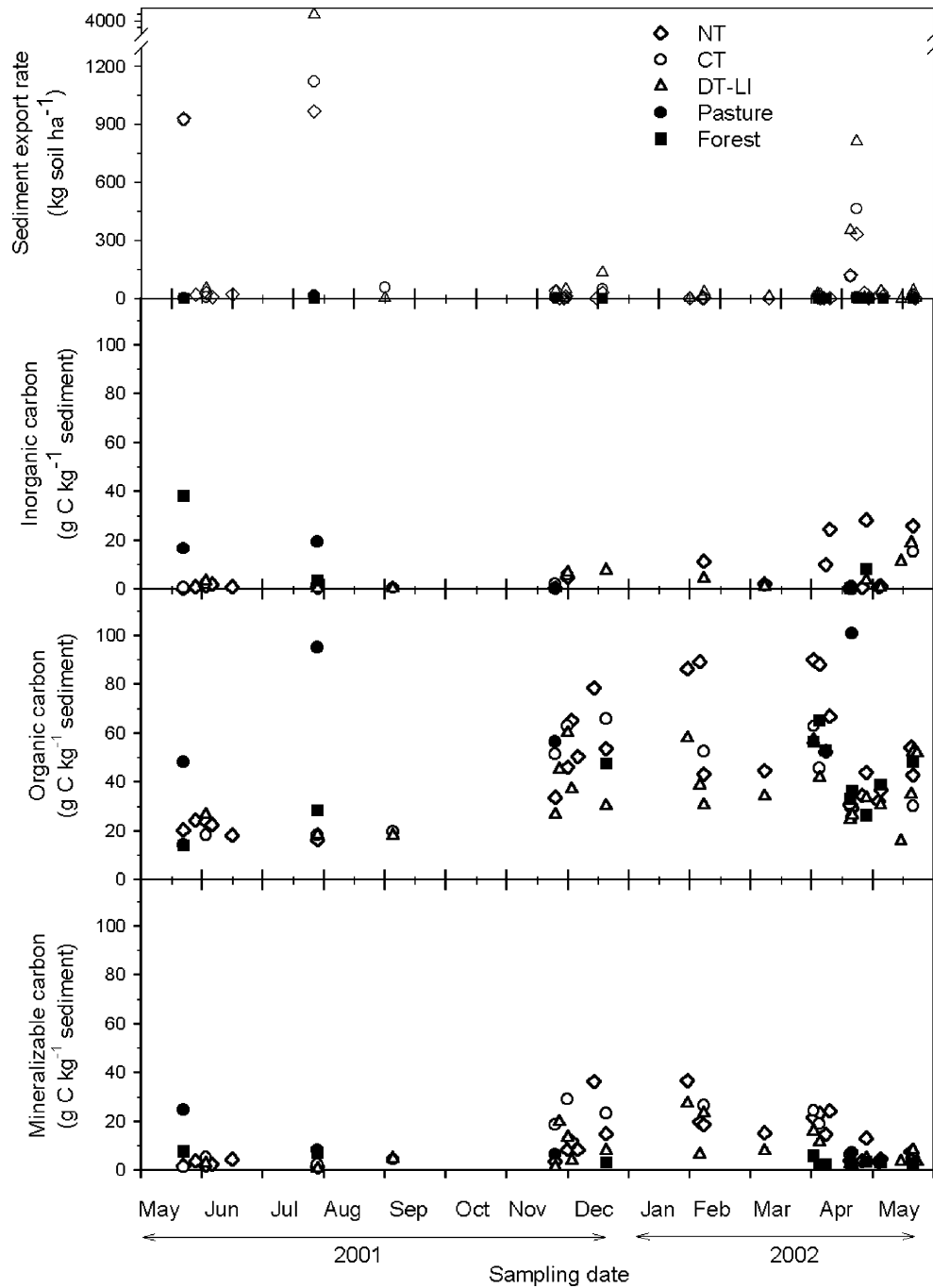
Fig. 1. Rates of export of soil, inorganic carbon, organic carbon and mineralizable carbon in runoff from cultivated and non-cropped watersheds at the North Appalachian Experimental Watershed during the period May 2001–May 2002.
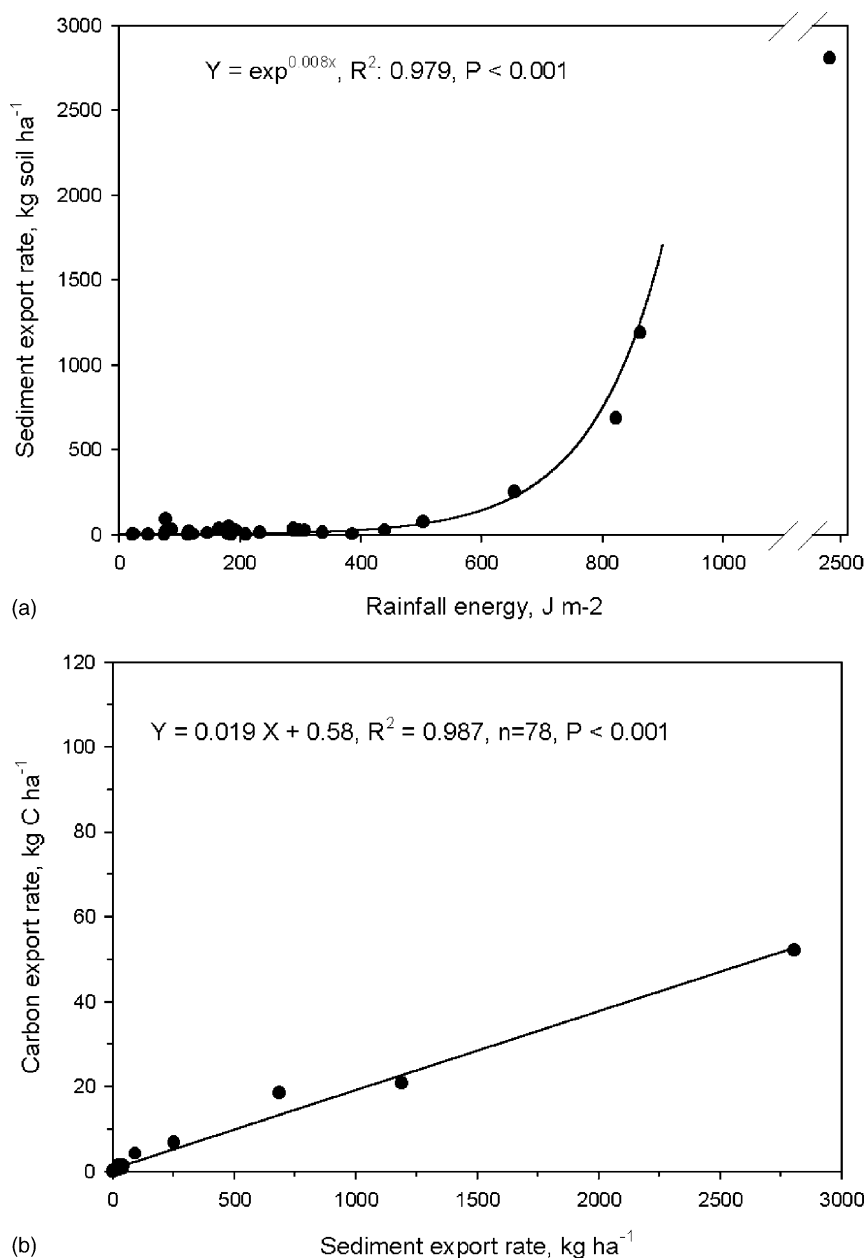
Fig. 2. Relationships between (a) daily soil loss and total rainfall energy and (b) soil loss and C export from watersheds.

significant effect of rainfall class was found. Dissolved organic C accounted for 11–28% of the C exported from the cultivated watersheds. In the non-cropped watersheds, most (67–76%) of the C loss was as dissolved load in runoff.

3.4. Biodegradability of runoff carbon

Carbon dioxide evolution during runoff incubation is presented in Fig. 3 for selected classes 1, 2 and 3 storms. In almost all cases, the CO_2 evolution data
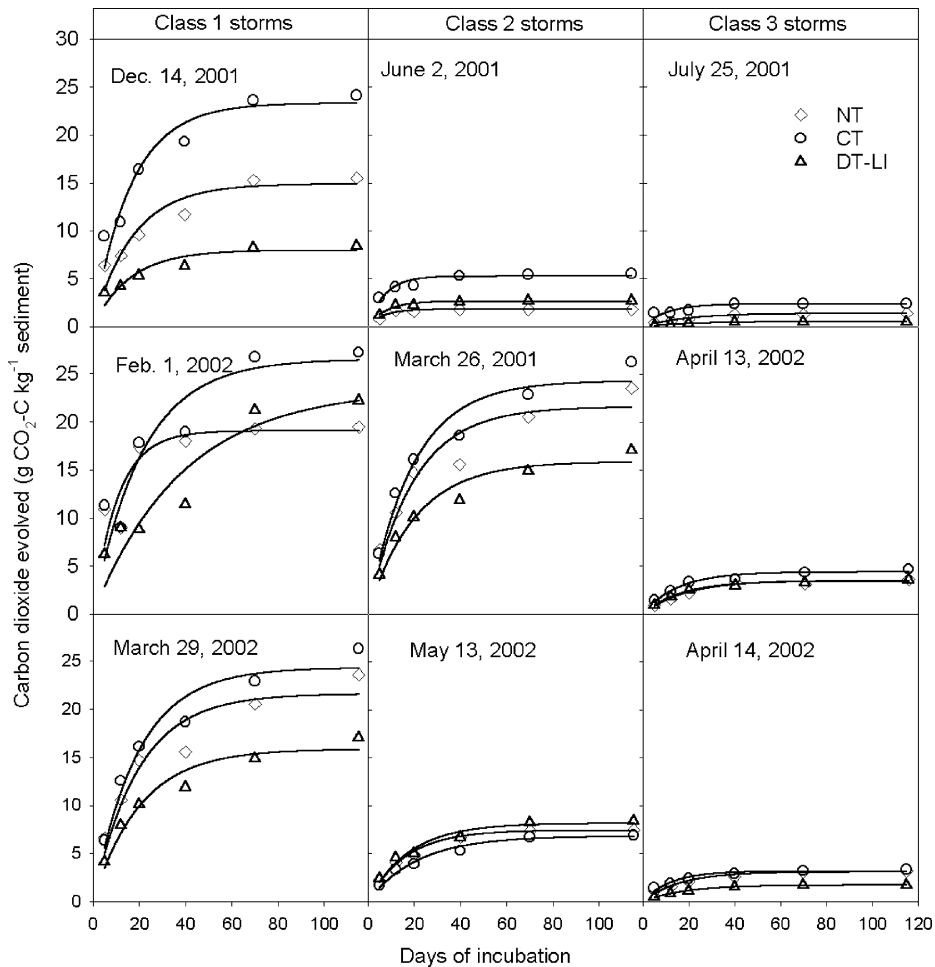
Fig. 3. Carbon dioxide evolution during incubation of runoff from cultivated watersheds following selected class 1 (left), class 2 (center) and class 3 (right) storms. Lines represent the best fit of the data to the first-order model.

fitted the first-order decomposition model very well. Across rainfall types and watersheds, the instantaneous decomposition rate constant (k) averaged 0.07 per day underscoring the short mean residence time ($1/k = 14$ days) of the C transported in overland flows. Among the cultivated watersheds, no effect of management was found with respect to the concentration of potentially mineralizable C (C_0) in runoff (Fig. 1). However, rainfall type had a significant effect ($P < 0.001$) on C_0 which averaged 14.9, 9.2 and 2.9 g C kg^{-1} sediment produced during classes 1, 2 and 3 storms, respectively (Fig. 3 and Table 4).

Consequently, and similar to the pattern observed with sediment organic C concentration (Table 4), C_0 was 1.5–2 times lower during the summer than in the other seasons (7.2 versus 14.6 g C kg^{-1} sediment). Dissolved organic C in runoff and C_0 were not correlated. Linear relationships (Fig. 4) were found between C_0 (Y) and total organic C concentration of eroded material (X). The slopes of the relationships indicate that, under classes 1 and 2 rainfalls, 30–40% of the eroded C could undergo mineralization; the mineralization potential was less (13%) for materials mobilized during class 3 storms.

Table 4
Organic and mineralizable carbon concentrations of sediment from cultivated watersheds as related to storm classes

| Land use and management practices	Storm classes					
	1		2		3	
	Organic C (g C kg^{-1} sediment)	MinC[a] (g C kg^{-1} sediment)	Organic C (g C kg^{-1} sediment)	MinC (g C kg^{-1} sediment)	Organic C (g C kg^{-1} sediment)	MinC (g C kg^{-1} sediment)
NT (WS 118)[b]	58.1 (5.8)[c]	17.6 (3.0)	44.7 (6.8)	7.4 (1.9)	22.9 (3.0)	2.9 (0.6)
CT (WS 123)	49.1 (5.8)	18.8 (3.3)	48.0 (14.9)	19.6 (7.3)	21.7 (2.9)	3.5 (0.9)
DT-LI (WS 127)	38.1 (3.9)	10.8 (2.7)	41.0 (5.8)	7.2 (2.4)	21.8 (2.1)	2.4 (0.9)
Pasture (WS 129)	56.4 (7.4)	7.4 (7.4)	n.d.	n.d.	81.2 (16.7)	25.3 (2.9)
Forest (WS 172)	48.0 (6.3)	17.9 (5.7)	47.7 (8.9)	13.0 (4.1)	27.9 (4.9)	10.1 (5.0)

[a] MinC: potentially mineralizable C.
[b] NT: no till; CT: chisel till; DT: disk till; LI: low chemical input; WS: watershed.
[c] Values are means with standard deviations in parentheses.
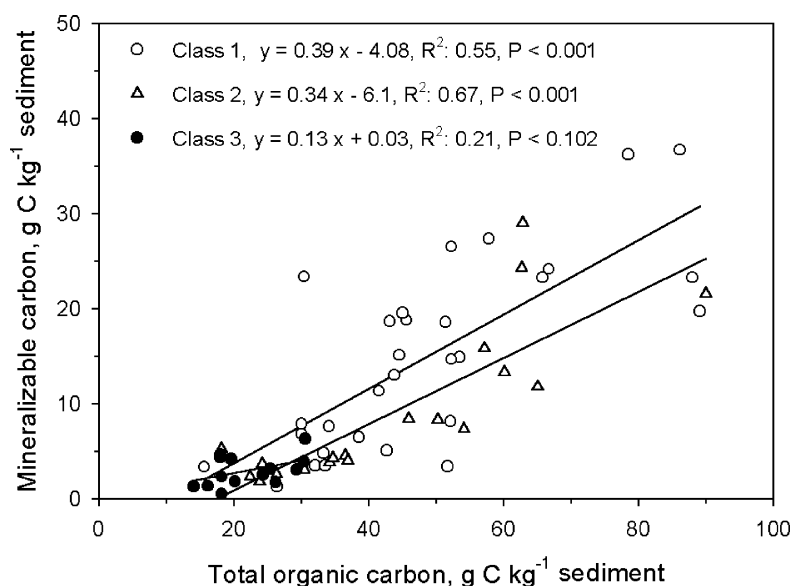
Fig. 4. Relationship between organic C and mineralizable C in eroded materials from cultivated watersheds.

4. Discussion

Soil and organic C losses from the DT-LI watershed was twice as large compared to the other practices. These results could be attributed to tillage frequency and crop cover. A fundamental rationale for initiating the DT-LI management practice in 1990 was to test the viability of disking and cultivation as alternatives to herbicides for the control of weeds. To that end, besides disking in the spring for seed-bed preparation, this watershed is typically cultivated at least twice during the growing season. This greater frequency of topsoil disturbance has probably contributed to the high soil and C losses recorded in the DT-LI watershed. It is also important to note that soil loss (6.2 Mg soil ha^{-1} per year, Table 1) in the DT-LI watershed during the 2001–2002 season was much higher than the 9 years average (1.15 Mg soil ha^{-1} per year) reported by Owens et al. (2002) but was less than soil loss (17.5 Mg soil ha^{-1}) from this watershed

recorded in 1969 (Shipitalo and Edwards, 1998). As noted earlier, the highly erosive rainstorm of 25 July 2001 (Fig. 1) had probably exacerbated soil losses during the 2001–2002 study period. The impact of such rare climatic events on erosional soil loss at the study site (Edwards and Owens, 1991; Shipitalo and Edwards, 1998) and elsewhere (Evans, 1995; Larson et al., 1997) is well documented. A few (5 out of 4000) but highly erosive rainfall events accounted for more than 60% of the cumulative soil loss recorded at the NAEW during a 28-year period (Edwards and Owens, 1991; Shipitalo and Edwards, 1998). Further, as soybean was grown in the DT-LI (and corn in the others) watershed during the period of observation, soil cover may have also contributed to the high soil loss. This interpretation is in accord with data from Owens et al. (2002) that have consistently shown more soil loss during the soybean phase of the various rotations being evaluated at the NAEW. Despite higher soil loss (hence C loss), SOC pool in the surface layer of the DT-LI watershed was similar to that of the NT and CT treatments. Losses of SOC from the DT-LI treatment appear to have been balanced by the annual addition of 5–9 Mg ha^{-1} of cattle manure.

Rainfall characteristic had a marked effect on sediment C concentration and the biodegradability of eroded C. As expected, the high-intensity convective (class 3) storms produced more sediment and consequently resulted in more C erosion rates compared to the low-intensity advective (class 1) storms. What was not expected, however, was the observation of an inverse relationship between rainfall energy, and both sediment C concentration and biodegradability (as determined by C_0). This observation is consistent with a previous report (Owens et al., 2002) from the study site documenting lower sediment C concentrations during the summer months. But Owens et al. (2002) did not investigate relationships between the observed temporal pattern of sediment C concentration and rainfall characteristics. The data presented in this paper show that relationships exist between rainfall energy and eroded C concentration and biodegradability. This is an important finding that, to our knowledge, has not been reported before and deserves further investigation. One could speculate that the energy loads of the low-intensity storms are sufficient for the detachment of the less resistant soil macroaggregates and the release of associated C. Further, the low overland flow rates that accompany such storms would favor the deposition of coarser and denser particles near the detachment points, and consequently lead to a progressive enrichment of the runoff in lighter and labile C fractions. Another explanation could be the transport, during larger storm events, of coarser materials or sediments containing less carbon originating from the bottom of rills several centimeter deep in the soil profile. Clearly, detailed and controlled experiments are needed to evaluate these and other conceptual mechanisms in order to elucidate the linkage between rainfall energy and the quality of C mobilized during erosional events. These additional studies, if conducted across a range of climatic regions and soil types, will further our understanding of the processes involved in C redistribution and improve our ability to assess the impact of water erosion on the global C cycle.

In contrast with the results of the microplot study (Jacinthe et al., 2002), data from this field investigation does not support the hypothesis of a relationship between soil C and runoff C quality. It appears however that at the watershed-scale level rainfall characteristics and hydrodynamic processes (in particular the sorting of particles during transport) dominate any effect that soil management practices may have on runoff C quality. Although this paper emphasizes rainfall characteristics to explain the variability in the biodegradability of runoff C, it does not however exclude the contribution of other processes such as changes in soil cover conditions and soil biochemical attributes. Previous studies (Franzluebbers et al., 1994; Campbell et al., 1999) reported a general increase in mineralizable C in soil during the growing season in cropped systems and attributed these variations to inputs of crop residues and rhizodeposition. In the present investigation, an increase in C_0 (Fig. 1) was noted during the fall (beginning with the 19 November 2001 storm and continuing until April). Beside the dominance of low-intensity rainfalls during that period, a greater concentration of labile C in the eroded sediment may be due to change in soil surface condition. Surface residue left after crop harvest in October may have contributed to an increased availability of fresh and decomposable materials.

Enrichment ratios (E_R = sediment C/soil C) of organic C in eroded material observed in this (Table 3)

and other studies (Palis et al., 1997; Jacinthe et al., 2002; Owens et al., 2002) typically ranged between 1.5 and 3. However, if the pool of mineralizable C in runoff and in soil is compared (Fig. 1 and Table 3), E_R values ranging between 18 and 40 are recorded (assuming that eroded soils originate from the top 10 cm soil layer). These E_R values for MinC are within the range observed with NT (29.5) and CT (37.8) soils in a simulation study using microplots (Jacinthe et al., 2002). These ratios show that 10–20 times more labile than total organic C fractions are removed by water erosion, and illustrate the high selectivity of erosion processes for the biologically active soil C pools.

The mass of labile C exported from the NT, CT and DT-LI watersheds amounted to 8.0, 10.7 and 10.8 kg C ha^{-1} per year, respectively. Between 54 and 74% of these amounts are associated with the high-energy storms. The total amount of organic C mobilized by the erosion process and redeposited within the watershed is probably several fold greater, given that soil export typically account for only 10% the soil displaced over an eroding landscape (Walling, 1983). Assuming that this delivery ratio is appropriate for the study site, the amount of labile C mobilized would average 100 kg C ha^{-1} per year, representing the additional soil C pool that could potentially be converted into atmospheric CO_2. This rate is lower than the 250 kg C ha^{-1} per year proposed previously through a modeling approach (Jacinthe and Lal, 2001).

5. Conclusions

This study investigated the influence of management practices and rainfall characteristics on erosional soil C loss and on the biological quality of eroded C. Soil and SOC losses were proportional to rainfall energy, and were substantially higher in the DT-LI treatment, a management practice involving frequent mechanical cultivation of the soil surface for weed control. The study showed that C mobilized during low-intensity rainfall was 2.5 times more labile than C released during high-energy storms. However, since more than 75% of the C mobilized during a cropping season occurs during the high-intensity storms, these rainfall events would most likely determine the

overall impact of water erosion on the cycling of terrestrial C.

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